

WATERBIRD RESPONSE TO CONSERVATION GRAZING IN  
WESTERN MINNESOTA TALLGRASS PRAIRIES

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## Abstract

Conservation grazing is becoming a popular management tool in prairie systems that evolved with grazing ungulates. Whereas it has been found to be an effective tool for maintaining upland prairie cover, its impacts on wetlands and the species that rely upon wetlands is not well-studied. Our objective was to investigate the impacts of conservation grazing on 2 groups of waterbirds: ground-nesting waterfowl and secretive marsh birds. As an economically important group of birds, much of the public land in Minnesota is managed for the production of waterfowl. Meanwhile, secretive marsh birds are believed to be an indicator species of wetland health, and their response to grazing could tell us how the whole wetland system is responding to management. In 2017 and 2018, we conducted waterfowl pair and brood counts and call-response surveys for secretive marsh birds in the tallgrass prairie of western Minnesota. We estimated abundance of blue-winged teal (*Spatula discors*) and mallard (*Anas platyrhynchos*) pairs and broods in response to grazing frequency and years since a site was last grazed. We used generalized linear models to estimate pair response and generalized linear mixed effects models to estimate brood response. Grazing had no effect on blue-winged teal pair abundance. Years since grazed had a weak, positive effect on mallard pair abundance, though increasing grazing frequency had no significant effect. Insufficient observations of mallard broods prohibited analysis of brood abundance, but we combined all dabbling broods together to estimate overall brood abundance in relation to grazing. Neither grazing variables influenced blue-winged teal or combined dabbling brood abundance. Using robust design occupancy models in Program MARK, we estimated marsh bird response to the same two grazing variables. American bittern (*Botaurus lentiginosus*),

sora (*Porzana carolina*), Virginia rail (*Rallus limicola*) all responded positively to at least one grazing variable, whereas pied-billed grebe showed no response. Our study suggests that the use of frequent, short-duration grazing to manage upland cover neutrally or positively impacts occupancy of secretive marsh birds and abundance of ground-nesting waterfowl pairs and broods.

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## **INTRODUCTION**

Until fairly recently, a widely-held belief was that grazing by livestock provided few benefits to prairie systems (Schieltz and Rubenstein 2016). This viewpoint likely stemmed from the harmful effects caused by unsustainable grazing practices during recent centuries (Briske et al. 2011). Year-round grazing at a high intensity led to a homogenous landscape that inhibited growth and diversity of native plant species (Briske et al. 2011, Richmond et al. 2012, Morris and Reich 2013, Minnesota Prairie Plan Working Group 2018). Overgrazing also negatively impacts wetlands, resulting in increased erosion and sedimentation, eutrophication, and the removal of emergent vegetation (Kantrud 1986, Harrison et al. 2017). However, in the tallgrass prairie of Minnesota, native prairie plants evolved in an ecosystem with high levels of ungulate grazing pressure (Samson and Knopf 1994, Grace 1998, Heisler et al. 2003), and sustainable grazing practices might promote structural diversity of upland and wetland vegetation to benefit a greater diversity of wildlife species (Kantrud 1986, Morris and Reich 2013).

North American prairies evolved with frequent disturbances that helped maintain their structure, including fire and grazing (Collins 1990, Fuhlendorf and Engle 2004). Regular wildfires inhibited fire-intolerant species, notably trees and shrubs, from becoming established themselves in prairies (Anderson 1990, Grace 1998). Fires burned unevenly, creating a mosaic of successional stages on the landscape that supported greater species diversity (Collins 1990, Fuhlendorf and Engle 2004). Recently burned patches were preferred by grazing ungulates, who further disturbed and shaped vegetation communities (Collins and Gibson 1990, Minnesota Prairie Plan Working Group 2018).

Additionally, both fire and grazing played an important role in controlling accumulation of dead plant litter, which decreases primary productivity (Collins 1990).

Although it is now widely recognized that frequent disturbance plays a critical role in shaping prairie ecosystems, humans spent the last 200 years removing most disturbance from North American prairies. Humans have eradicated some native North American grazing ungulates, most notably bison (*Bos bison*), and greatly reduced the abundance of others such as pronghorn (*Antilocapra americana*) and elk (*Cervus canadensis*). In addition, human suppression of wildfires because they pose a significant threat to the livelihood and survival of those living and working on the land (Collins 1990, Kantrud 1986), has changed the vegetation structure and composition in prairies. In the absence of disturbance, trees, shrubs, and non-native and invasive species have infiltrated prairie systems, out-competing native prairie species (Samson and Knopf 1994, Grace 1998, Bakker 2003, Heisler et al. 2003, Minnesota Prairie Plan Working Group 2018).

This widespread degradation of North American prairies has occurred in conjunction with dramatic losses of prairies and prairie wetlands. An estimated 99.6% of Minnesota's native tallgrass prairie has been lost since European settlement (Samson and Knopf 1994, Minnesota Prairie Plan Working Group 2018) as a direct result of widespread cultivation and urban development (Samson and Knopf 1994, Thompson et al. 2016). Factors that influence Minnesota's prairies also influence wetlands embedded within them. In the last 200 years, 91.9% of wetlands in Minnesota's Prairie Parkland Region have disappeared, primarily to drainage due to agricultural practices (Minnesota Prairie Plan Working Group 2018). Greater than 50% of remaining wetlands are rated to

be in poor condition by the Minnesota Pollution Control Agency (Genet 2015). This loss in quality is due in large part to the proliferation of invasive emergent vegetation (Genet 2015, Minnesota Prairie Plan Working Group 2018), which have been able to establish themselves in the absence of regular disturbances (Minnesota Prairie Plan Working Group 2018). Undisturbed, shallow wetlands quickly become dominated by invasive hydrophytes, particularly hybrid cattails (*Typha X glauca* Godr. and *T. angustifolia*) and reed canary grass (*Phalaris arundinacea*; Kantrud 1986, Genet 2015, Minnesota Prairie Plan Working Group 2018). Hybrid cattails form monocultures that dominate entire wetlands, leaving little open water and negatively impacting native plant species and structural diversity (Genet 2015).

Because wildfires pose a significant threat to human infrastructure and agricultural crops on the landscape, they continue to be actively suppressed. And with so many of our native grazers gone, remnant prairies would go undisturbed without human intervention. In the absence of natural disturbances, land managers often attempt to mimic the effects of historical fire and grazing regimes with the objective of restoring and maintaining native prairie conditions in remnants of former, extensive prairie systems in Minnesota. Notable objectives of mimicking historical disturbance include suppression of woody and non-native species, encouragement of growth of native plant species, reduction of litter buildup, and promotion of a structurally diverse plant community that can promote a diverse wildlife community (Collins 1990, Devries and Armstrong 2011). Whereas prescribed burning is widely-regarded as an effective disturbance tool in prairies (Collins and Gibson 1990), it is expensive, logistically complicated, and difficult to perform at the necessary intensity and frequency (Thompson et al. 2016). Because woody

vegetation can benefit from infrequent burning (Grace 1998, Heisler et al. 2003), it is important that land managers consider other disturbance tools if they are unable to perform burns as frequently as necessary to achieve desired outcomes.

One potential alternative to burning is conservation grazing (Minnesota Prairie Plan Working Group 2018), which attempts to mimic the widespread grazing formerly conducted by bison and other indigenous ungulates (Grace 1998, Heisler et al. 2003, Fuhlendorf and Engle 2004). Livestock grazing can positively impact upland cover, controlling dead litter buildup, promoting native species growth, and creating structural diversity of prairie plant communities (Pederson 2006, Morris and Reich 2013). However, the impacts of conservation grazing on wetlands and wetland-dependent wildlife are less well-known (U.S. Fish and Wildlife Service 2017). Continuous grazing at intensive levels is recognized as being harmful to wetland health (Kantrud 1986), but conservation grazing performed at more sustainable levels might not negatively affect or may potentially positively affect wetland quality (Minnesota Prairie Plan Working Group 2018). If conservation grazing is going to be used by land managers in the tallgrass prairies of the Midwest, it is important to consider how grazing impacts prairie wetlands, and the species dependent on those wetlands.

Ground-nesting waterfowl rely on both uplands and wetlands during the breeding season. Upland cover types serve as nesting habitat (Reynolds et al. 2001, Stephens et al. 2005), whereas wetland cover types with emergent vegetation provide brood-rearing habitat (Raven et al. 2007). However, extensive, dense emergent cover may reduce the attractiveness of prairie wetlands as brood-rearing habitat to some dabbling duck species (Walker et al. 2013). Because waterfowl management is a conservation priority on many

public lands, determining how ground-nesting ducks respond to grazing will likely be important to land managers considering conservation grazing as a management tool. The effect of grazing on waterfowl is not clear, in that both positive and negative effects have been observed in previous investigations (Duebbert et al. 1986, Warren et al. 2008, Harrison et al. 2017). Furthermore, the majority of these studies have focused on the impacts of grazing by comparing grazed to ungrazed sites without considering other factors (i.e., time of year, intensity, or frequency; Briske et al. 2011, Schieltz and Rubenstein 2016). The few studies that have investigated waterfowl breeding success across a range of grazing intensities have found benefits at low-to-medium levels of grazing, but negative effects at high levels (Warren et al. 2008, Harrison et al. 2017). Although we are unaware of any studies that have investigated how the frequency of grazing affects waterfowl productivity, how waterfowl respond to frequency of disturbance might be as important as intensity of disturbance (Devries and Armstrong 2011).

Secretive marsh birds include rails, bitterns, and grebes, and although they are not closely related taxonomically, they all rely on wetlands with a juxtaposition of dense emergent vegetation and open water (Lor and Malecki 2006). But the accumulation of dense, emergent vegetation can impede the ability of waterbirds, including secretive marsh birds, to move and forage (Johnson 1984, Ma et al. 2010). Additionally, monocultures of hybrid cattail create a structurally homogenous vegetation community that is unattractive to many waterbirds (Kantrud 1986, Minnesota Prairie Plan Working Group 2018). Occupancy of secretive marsh bird is believed to be negatively associated with the invasion of nonnative emergent vegetation, and their diet of aquatic invertebrates

makes them vulnerable to bioaccumulation of contaminants, suggesting that they are a potential indicator species of wetland health (Conway 2011). Monitoring secretive marsh bird populations as they relate to wetland management practices could provide insight into how the entire system is reacting to manipulations.

Our objective was to evaluate occurrence and abundance of waterbirds in relation to conservation grazing in the tallgrass prairie of western Minnesota. Collectively, the U.S. Fish and Wildlife Service (USFWS), Minnesota Department of Natural Resources (DNR), and non-profit conservation organizations such as The Nature Conservancy (TNC) manage thousands of acres of remnant and restored native prairie throughout western Minnesota (Johnson 1997) and conservation grazing is a potentially valuable tool for managers who are unable to burn as often as necessary to mimic historical disturbance regimes. With so few of the state's original prairie wetlands still present, and even fewer of good quality, it is important to understand how conservation grazing practices might affect a vulnerable group of birds that rely on wetlands and emergent vegetation during a critical time in their life cycle.

## **STUDY AREA**

We surveyed wetlands on U.S. Fish and Wildlife Service (USFWS) Waterfowl Production Areas (WPAs) in the Morris Wetland Management District (WMD) in western Minnesota. Historically, western Minnesota was dominated by an extensive tallgrass prairie ecosystem system that provided important breeding and stopover habitat for migrating birds. Since European settlement, the landscape of western Minnesota and much of the tallgrass prairie ecosystem has largely been converted to row-crop agriculture with concomitant loss of wetlands (U.S. Fish and Wildlife Service 2012).

Within the Morris WMD, ~10% of the historical wetlands and ~1% of native prairie remain intact (U.S. Fish and Wildlife Service 2012, 2017). Despite these dramatic losses in extent of prairies and wetlands, this region's remnant prairies and wetlands still provide important habitat to many species of migratory birds.

The USFWS manages WPAs primarily for the production and conservation of migrating waterfowl (U.S. Fish and Wildlife Service 2017). The Morris WMD is in Minnesota's Prairie Parkland Region and lies along the eastern edge of the Prairie Pothole Region (PPR). The Morris WMD is characterized by a landscape transitioning from relatively flat tallgrass prairie in the west to rolling hardwood forests in the east (U.S. Fish and Wildlife Service 2017). Wetlands in the western portion of the WMD are smaller and shallower with more emergent vegetation, whereas wetlands in the eastern counties are larger and deeper with more open water.

A primary management objective of the Morris WMD is to maximize waterfowl production by focusing on upland habitat enhancement. Managers actively burn, graze, and hay properties under their control to promote growth of native prairie plants and combat woody and nonnative species encroachment (U.S. Fish and Wildlife Service 2017). The Morris WMD has an extensive grazing program and collaborates with cattle producers by providing use of pastures in exchange for a grazing fee. Their grazing objectives include reducing accumulation of biomass, engineering a more heterogeneous landscape, and promoting the growth of native grasses through reduced competition from invasive species (U.S. Fish and Wildlife Service 2017). To achieve these objectives, they focus efforts on short-duration, high intensity grazing with at least one year of rest. Grazing contracts typically stipulate 30-day grazing periods between 1 May and 31 July



at a stocking rate of 0.3 animal unit month (AUM)/hectare (1 AUM/ac; U.S. Fish and Wildlife Service 2017).

## **METHODS**

### **Site Selection**

We selected sites with a variety of grazing histories to investigate secretive marshbird and dabbling duck response to grazing. Larger WPAs were often divided into smaller management units with variable management histories. In these instances, we often had > 1 site on a WPA. Using WPA management histories, we grouped sites by the number of years since they were last grazed: 0 (currently being grazed), 1, 2, 3, 4, 5, 6, or 7+ years. To address grazing frequency, we also grouped sites by the number of times they had been grazed in the 3 years prior to being surveyed: 0, 1, 2, or 3.

Due to the limited number of sites with grazing as the only form of recent disturbance management, selecting sites randomly was not possible. Because the district actively manages with fire and grazing, approximately one third of sites had been burned and grazed in the previous 6 years. We excluded 16 sites from our 2018 sample because they were mowed or burned between field seasons or planned to be burned during the 2018 field season and replaced them with 18 previously unsurveyed sites. These new sites became available in 2018 for one of two reasons: 1) previously ungrazed sites were grazed in 2017 or planned to be grazed during the 2018 field season, or 2) recently grazed sites became available when we increased the distance we were willing to travel in 2018.

### **Ground-nesting Waterfowl**

*Field methods.*— We conducted 4 rounds of waterfowl surveys in 2017 and 2018. During the first 2 rounds, our sampling methods were influenced by concurrent surveys

of secretive marsh birds, which are performed in the mornings and evenings. Given that past studies have endorsed both morning and evening waterfowl surveys (Pagano and Arnold 2009a), we conducted visual surveys for waterfowl pairs in the morning (visible light – 1030) and evening (1700 – dark). We conducted pair surveys from 12 May – 3 June in 2017 and 15 – 29 May in 2018. We missed the optimum window for pair counts of early-nesting waterfowl (i.e. mallard [*Anas platyrhynchos*], northern pintail [*Anas acuta*]) in 2017; however, a much colder than average April and late ice out (Waterfowl Population Status 2018) may have pushed back that window for early-nesters in 2018. The final 3 rounds of waterfowl surveys, conducted between the beginning of June and the end of July, functioned as brood surveys. Because we were no longer surveying for marsh birds during the third and fourth visits, we conducted the last 2 rounds of brood surveys during the morning. To estimate detection probabilities for broods we conducted same-day replicate surveys during the third round of surveys. We completed replicate visits from 1 – 4 hours after the first visit.

To count pairs and broods at each site, we conducted visual surveys of every wetland with visible open water, omitting wetlands in which we could observe no open water because emergent vegetation either completely covered the wetland or obscured view of open water because the probability of observing waterfowl in those wetlands was near zero. We alternated observers between visits to minimize impacts of observer variation in our models. At the beginning of each survey, observers estimated ( $\pm 10\%$ ) the proportion of a survey wetland covered by emergent vegetation using both aerial images of the wetland and various vantage points around the wetland. They also recorded an estimate ( $\pm 25\%$ ) of the proportion of a wetland basin holding water (wetland inundation).

When observers detected waterfowl, they recorded species and social category: lone pair, lone male, lone female, grouped pairs, group of  $\leq 5$  males, group of  $> 5$  males, grouped females, group of 2 males and 1 female, other mixed-sex groups, and broods (Dzubin 1969). When observers detected broods, they counted the number of ducklings and estimated their age (Gollop and Marshall 1954).

To ensure observers were able to competently survey an entire wetland, we excluded wetlands  $> 5$  ha. On vegetated wetlands and larger wetlands, observers used multiple vantage points to adequately survey an entire wetland, spending  $\geq 5$  minutes per wetland to maximize detection probability (Pagano and Arnold 2009a). We did not return to wetlands that were dry or completely occluded by vegetation. We halted surveys when wind exceeded 50 km/hour, precipitation persisted, or fog impeded our ability to see. If we were unable to completely survey a site surveyed due to inclement weather, we surveyed it again on the next available day.

*Statistical methods.*— To evaluate waterfowl pair and brood response to grazing, we used an *a priori* set of candidate models (e.g., Ahlering et al. 2018) to assess whether the inclusion of grazing variables improved model performance beyond a base model. We built base models for pair and brood abundance that included site- and landscape-level characteristics that have been associated with dabbling duck pair and brood abundance in prior studies. The site-level covariates were: 1) longitude, 2) wetland inundation, 3) percent of emergent vegetation covering the wetland, 4) area of surveyed wetland, 5) wetland permanence (Shaw and Fredine 1956), and 6) the number of years since the site was last burned. The landscape-level covariates included were 1) the proportion of land cover classified as wetland habitat (National Wetlands Inventory

2018) within a buffer of the surveyed wetland (2,000-m for pairs and 500-m for broods; Harrison et al. 2017) and 2) the proportion of land cover classified as agricultural within those buffers (National Agricultural Statistics Service 2017). We also included observer and time of day because previous studies have shown that they can impact detection probability (Pagano and Arnold 2009). Last, we included survey day and year in our base model as additional covariates influencing abundance. Using z-transformation, we re-scaled our quantitative covariates prior to analysis.

We modeled pair abundance using generalized linear models with indicated breeding pairs of a particular species as our response variable. We used the social categories explained earlier to calculate indicated breeding pairs: lone pairs + grouped pairs + max(lone males or lone females) + 2M:1F + max(grouped males  $\leq 5$  and grouped females). To build our base brood abundance models, we fit generalized linear mixed effects models (glmer in lme4 package in R) with number of broods as our response variable and the surveyed wetland (Wetland ID) as a random effects variable. Our base models for pair and brood abundance included the list of independent variables described above. We then fit three grazing models for each set of analyses: 1 model with grazing frequency, 1 model with a linear variable for years since grazed, and to account for potential non-linear responses, we added a quadratic term for years since grazed to a third model. If a grazing model had a lower AIC value than its corresponding base model, we concluded that grazing had a demonstrable effect on waterfowl abundance.

### **Secretive Marsh Birds**

*Field Methods.*—We randomly selected survey wetlands using a random number generator. To minimize counting the same bird at multiple survey points, we spaced

points  $\geq 400$  m apart (Conway 2011). On larger sites that allowed for  $> 1$  wetland to be surveyed, we randomly selected a second wetland to survey. Following the protocol of Conway (2011), we placed survey points along the edges of selected wetlands where emergent vegetation was present. We grouped secretive marsh bird points into survey routes based on proximity and maintained routes and survey order across each field season (Conway 2011). We modified survey routes between years to account for sites that we excluded or added from 2017 to 2018.

We adapted the Standardized North American Marsh Bird Monitoring Protocol (Conway 2011) to survey 6 species of secretive marsh birds: American bittern (*Botaurus lentiginosus*), least bittern (*Ixobrychus exilis*), pied-billed grebe (*Podilymbus podiceps*), sora (*Porzana carolina*), Virginia rail (*Rallus limicola*), and yellow rail (*Coturnicops noveboracensis*). We surveyed each route twice per season, approximately 18 days apart (mean 17.7, range 10 – 26 days). We surveyed routes once in the morning (30 minutes prior to sunrise to 0930 hours) and once in the evening (1800 hours to 30 minutes after sunset; Bolenbaugh et al. 2011) using different observers for each visit.

Surveys were 11 minutes in duration, consisting of a 5-minute period of passive listening followed by 6 minutes of broadcasting common breeding and territorial calls of our 6 target species to elicit responses from secretive marsh birds. We obtained the standardized broadcast sequence from the North American Marsh Bird Monitoring Program Coordinator in the form of an MP3 file. The order of calling was Least Bittern, Yellow Rail, Sora, Virginia Rail, American Bittern, Pied-billed Grebe. Each minute of the broadcast sequence consisted of 30 seconds of one species vocalizing followed by 30 seconds of silence. Using a big game caller (Cass Creek Big Horn Remote Speaker)

connected to a phone (used as an MP3 player), we broadcasted calls at 90 dB 1 m from the speaker (Conway 2011). Upon hearing any of the focal species, the observer recorded the species, type of call, minute detected, whether the bird was seen and/or heard, and estimated location to help determine if a bird heard later in the survey was a different bird. Although the priority of this survey was secretive marsh birds, observers also recorded occurrence and abundance of other bird species seen or heard within a 100-m radius during the first minute during the survey they were detected.

At the beginning of each survey, observers estimated ( $\pm 10\%$ ) the proportion of a survey wetland covered by emergent vegetation using both aerial images of the wetland and various vantage points around the wetland. They also recorded an estimate ( $\pm 25\%$ ) of the proportion of a wetland basin holding water (wetland inundation). When wind impeded our ability to detect calls (wind  $\geq 24$  km/h; 15 mph), we halted surveys. Additionally, we paused or halted surveys in the event of continuous precipitation. If a survey route could not be completed on a single day, we started the survey route over on the next available day (Conway 2011).

*Statistical methods.*— We assessed the impact of grazing and other variables on occurrence of secretive marsh birds with robust design occupancy models (McKenzie et al. 2003) implemented in Program MARK (White and Burnham 1999) using a limited, *a priori* set of candidate models (Ahlering et al. 2018). Robust design occupancy allowed us to incorporate multiple visits (i.e., 2 visits per year, for 1 or 2 years) without assuming an individual site maintained the same occupancy status throughout all 4 surveys. Because our primary focus was on occupancy, we parameterized the model by estimating occupancy during all 4 survey periods ( $\psi_t$ ) and estimating local extinction ( $\epsilon_t$ ) between

years (McKenzie et al. 2003). We expected vocalization probabilities of each species to differ in their response to playback and occupancy to differ in response to grazing, and so we modeled each species separately.

We first modeled vocalization probability for each species using an intercept-only model for  $\psi_t$  and  $\varepsilon_t$  that considered how broadcasted calls impacted the probability of a species calling during each minute of a survey. We reasoned that there were 4 likely vocalization behaviors in response to broadcasted calls, all of which assumed a constant detection probability throughout each minute of the initial 5-minute passive listening period. First, we considered that the broadcasted calls did not influence the probability of a species vocalizing; therefore, the model had a constant detection probability throughout the 11-minute survey period. Second, we considered that only conspecific calls influenced vocalization probability. Thus, we held detection probability constant through periods of passive listening and all minutes during which we broadcasted calls of other species while allowing a different (presumably greater) detection probability during the minute when we broadcasted conspecific calls. The third and fourth models considered that both conspecific and heterospecific calls influenced vocalization probability. The third model included 1 vocalization probability for all minutes during which we broadcasted heterospecific calls and 1 for the minute we broadcasted conspecific calls, and the fourth model allowed for variation in vocalization probability among all 6 minutes during which we broadcasted calls. Using second-order Akaike's Information Criterion ( $AIC_c$ ), we determined the best performing model of each species' response to broadcasted calls.

We then built a base model that incorporated a suite of covariates that have been documented in past studies to impact detection probability and occupancy. We included 6 covariates that potentially influence detection probability: 1) observer (using 2 dummy covariates to distinguish among 3 observers over 2 years), 2) noise level (ordinal scale from 0-4, Conway 2011), 3) AM vs. PM, 4) time since sunrise/sunset, 5) survey day, and 6) year. Because the primary objective of our project was to investigate the impact of grazing on occupancy of secretive marsh birds, our analysis was designed to assess whether the addition of grazing variables improved model performance beyond a base occupancy model that included influential site- and landscape-level covariates. Accordingly, our base model included site- and landscape-level characteristics that past studies have shown to be correlated with marsh bird occupancy. The site-level characteristics we included were: 1) area of the surveyed wetland, 2) percent of emergent vegetation covering the wetland, 3) wetland permanence class (Shaw and Fredine 1956), and 4) the number of years since a site was burned (range 0-30). We also included the total wetland area within a 200-m buffer as a landscape-level covariate. To calculate this last covariate, we created 200-m buffers around the central points of surveyed wetlands and overlaid all wetland types intersecting the buffers. We then calculated the total area in hectares of wetlands intersecting each buffer. All detection and occupancy covariates were internally standardized (mean 0, SD 1) in Program MARK prior to analysis.

Finally, to assess whether marsh bird occupancy was affected by grazing, we fit 3 additional models that included grazing covariates. Although we selected survey sites based on the number of years since a site was grazed (years since grazed), we created an additional grazing covariate to represent grazing frequency: the number of times a site



had been grazed in the last 3 years. Because these 2 grazing covariate were highly correlated ( $r = -0.745$ ), we considered their effects in separate models. We fit 1 model with grazing frequency, 1 model with a linear covariate for years since grazed, and to account for potential non-linear responses, we added a quadratic term for years since grazed to a third model. We compared the 3 grazing models to the base model described above using  $AIC_c$ .

### **Emergent Wetland Vegetation**

To assess the potential effects of grazing on emergent wetland vegetation, we estimated percent (0-100) emergent wetland vegetation (to the nearest 10%) covering our study wetlands using aerial imagery and various vantage points around a wetland. We fit linear models with percent emergent vegetation as the response variable and grazing as an indicator variable. We considered our 2 grazing variables (years since grazed and grazing frequency) in separate models, as we did for our marsh bird analysis. Recognizing that there was likely a relationship between percent emergent wetland vegetation and wetland permanence, we fit a third model with wetland permanence class (Shaw and Fredine 1956) as a predictor variable. We combined wetland classes 1 and 2 (ephemeral and temporary, respectively), as there were insufficient numbers of each to analyze separately. Last, we fit models with an additive and an interactive effect between permanence class and grazing variables. With each of our grazing variables, we fit 1 model with grazing and wetland class as predictor variables and 1 model with an interaction between grazing and wetland class. We compared the  $AIC$  values of the 7 models (Table 1) to identify the best-supported model and determine if grazing had any effect on percent emergent wetland vegetation.

## RESULTS

We surveyed 70 sites, including 16, 18, and 36 sites surveyed in 2017, 2018, or both years, respectively. Between 2017 and 2018, we surveyed 89 sites that had been grazed in the 6 years prior to being surveyed and 17 sites that had not been disturbed in  $\geq 7$  years (Fig. 1a). Within our grazing frequency category, 38 sites had not been grazed in the 3 years prior to being surveyed, 34 sites had been grazed once, 31 sites had been grazed twice, and 3 sites had been grazed three times (Fig. 1b).

### Emergent Wetland Vegetation

The best-supported model of percent emergent vegetation included the additive effects of grazing frequency and wetland class (Table 1). Percent emergent wetland vegetation was negatively associated with increasing grazing frequency (beta = -6.613, SE = 1.519, Fig. 2a). The model with additive effects of wetland class and years since grazed was also competitive ( $\Delta AIC = 2.12$ ), indicating that recentness of grazing was also associated with percent emergent vegetation, with percent vegetation showing a positive relationship with years since grazed (beta = 2.667, SE = 0.651, Fig. 2b). There was some evidence that grazing effects varied among wetland classes, but the interaction effect was not strong enough to overcome the penalty for adding 4 additional parameters to AIC (Table 1).

### Ground-nesting waterfowl

We detected 916 blue-winged teal (*Spatula discors*), 705 mallard, 60 gadwall (*Mareca strepera*), 55 northern shoveler (*Anas clypeata*), and 17 green-winged teal (*Anas carolinensis*) indicated breeding pairs and 70 blue-winged teal, 30 mallard, 4 gadwall, and 13 unknown dabbling broods. Because we had relatively small sample sizes for most species, we analyzed pair abundance of blue-winged teal and mallard and brood

abundance of blue-winged teal and all dabbling ducks combined. Blue-winged teal broods comprised 58.9% of the dabbling group, whereas mallards, gadwall (*Mareca strepera*), and unknown dabbling broods made up 40.1%. The best-supported model for mallard pair abundance included a positive relationship with years since grazed (Table 2, Fig. 3b). Years since grazed was not included in the best-supported model of abundance of blue-winged teal pairs (Table 2, Fig. 3a), and grazing frequency was not included in the best-supported models of pair abundance of either species (Table 2, Fig. 4). Neither grazing frequency nor years since grazed was included in best-supported models of brood abundance of blue-winged teal or combined dabblers (Table 2).

### **Secretive Marsh Birds**

We conducted 350 marsh bird surveys and detected 579 individuals marsh birds, with soras detected on 76 sites, pied-billed grebes on 57, Virginia rails on 45, American bitterns on 25, least bitterns on 11, and yellow rails on 0 sites. We modeled detection probability and occupancy of soras, pied-billed grebes, Virginia rails, and American bitterns, but there were insufficient detections of least bitterns and yellow rails to construct models of detection probability or occupancy for these species.

*Vocalization Probability.*— The 4 species of marsh birds responded differently to broadcasted calls (Fig. 5). Vocalization probability of American bittern was constant across the 11-minute survey (Table 3). Pied-billed grebe vocalization probability was greatest following their own broadcasted calls and consistently low during both the initial passive listening period and during broadcasted calls of all other species (Table 4). The top model for soras (Table 5) and Virginia rails (Table 6) included 7 vocalization probabilities within a survey: vocalization remained consistently low during the passive

listening period and varied across the 6 minutes of broadcasted calls. Whereas the probability of soras vocalizing was the highest during their own calling minute, they had a higher probability of vocalizing following the sora calling minute than the minutes prior. Virginia rails exhibited a similar pattern of vocalization probability, although our results suggested that their probability of vocalizing only remained high for the one minute following its own calling minute.

*Occupancy.*— Grazing was included in the best-supported occupancy model of 3 species of marsh birds. Both years since grazed and grazing frequency improved model performance for American bittern (Table 3), though the model that included years since grazed had a lower AIC<sub>c</sub>, indicating that it was a better predictor of occupancy. Sites were less likely to be occupied by American bitterns the longer they had been since they were grazed (Fig. 6), and more frequently grazed sites were more likely to be occupied (Fig. 7). Virginia rails responded positively to increased grazing frequency (Fig. 7), and sora occupancy decreased in response to increasing years since grazed (Fig. 6). Neither grazing frequency nor years since grazed were related to pied-billed grebe occupancy (Table 4).

## **DISCUSSION**

Conservation grazing decreased the percentage of emergent vegetation in study wetlands. In addition, the longer the time since last grazed was positively related to percent of emergent vegetation in wetlands. However, our results show that conservation grazing at the intensity that occurred on our study site is unlikely to substantively alter the percent of emergent vegetation covering a wetland. A modest reduction of about 6.5% can be expected in response to trampling and grazing on emergent vegetation, exposing open

water and increasing interspersions of water and vegetation. A modest increase of about 2.5% emergent wetland vegetation was seen each year a site went ungrazed. Additional research into the impact of grazing frequency on water quality would give us more information on how wetlands respond to conservation grazing.

A modest decrease of emergent vegetation is likely appropriate for most secretive marsh bird species, that prefer wetlands dominated by vegetation (Gibbs et al. 1991, Lor and Malecki 2006). For American bitterns, site occupancy increased with frequency of grazing and decreased the longer wetlands remained ungrazed. Although little is known about the ecology of this secretive marsh bird, they prefer shallower wetlands with an interspersions of vegetation and open water (Lowther et al. 2009). Grazing that opened up heavily choked wetlands and increased interspersions of open water would likely be beneficial for these birds. Best-supported models of sora occupancy indicated that wetlands were more likely to be occupied when they experienced more frequent grazing and occupancy decreased the longer it had been since a site had been grazed. However, because our best-supported model of emergent vegetation indicated that grazing only modestly reduced wetland vegetation even when grazing occurred multiple years in a row, it is unlikely that 2 or 3 consecutive years of grazing reduces wetland vegetation cover to a degree that the probability of occupancy decreases. wetlands would become unattractive to Soras. Conversely, undisturbed wetlands become increasingly choked by dense vegetation and may become less attractive to soras. Although our 2 grazing variables were highly correlated, only grazing frequency was an important predictor of Virginia rail occupancy. This could be a result of their preference for earlier-successional wetlands that do not have an accumulation of residual standing biomass (Conway 1995).

Grazing multiple years within a short span likely increases the impact of trampling and grazing on emergent vegetation, more effectively controlling accumulation of biomass and maintaining a wetland that does not impede movement of Virginia rails.

Pied-billed grebes have substantively different habitat preferences than the other 3 species of marsh birds for which we were able to model occupancy. They prefer densely vegetated wetlands interspersed with a considerable amount of open water (Weller and Spatcher 1965, Bolenbaugh et al. 2011). Pied-billed grebes forage for fish and invertebrates and escape danger by diving and thus prefer wetlands with deeper water (Muller and Storer 1999, Baschuk et al. 2012). Deep-water wetlands are less likely to be dominated by invasive emergent vegetation, and it is therefore unsurprising that grazing frequency or years since grazed were not included in best-supported models of occupancy of pied-billed grebes. Because we observed a modest effect of grazing on percent emergent vegetation, it is possible that a longer duration or additional grazes within the same season could trample and consume perimeter vegetation that grebes rely upon for nesting cover. However, our results indicated that, when performed in the late spring and for a short duration, grazing frequency and years since grazed were not related to pied-billed grebe occupancy.

Grazing wasn't associated with abundance of blue-winged teal pairs, broods, or combined dabbling broods, suggesting that conservation grazing did not alter vegetation to a degree that affected wetland attractiveness for this species. Mallard pair abundance was positively associated with years since grazed. However, given the small effect size that increasing grazing frequency was not related to Mallard pair abundance, we do not believe that these results should dissuade land managers from using conservation grazing

as a disturbance tool. Our results suggest that short-duration, high intensity grazing at least once every 3 years to manage upland cover does not negatively affect pair or brood abundance of dabbling ducks.

Although our results indicated that increased grazing frequency was not negatively related to marsh bird occupancy or ground-nesting duck pair and brood abundance, a few caveats are in order. The tallgrass prairie is a more productive system than more arid prairies. Grazing at the intensity applied in this study could have a different impact in drier prairies such as mixed grass and short grass prairies, and so our results should be applied cautiously in more arid habitats. Additionally, our study only analyzed effects of short duration, high intensity grazing in the late spring, and we cannot assume similar results on the tallgrass prairie when different grazing prescriptions are applied.

### **Management Implications**

Management of restored and native prairies frequently focuses on upland management (U.S. Fish and Wildlife Service 2017). But in prairie systems like the Prairie Pothole Region, wetlands provide habitat for many species, and upland management could have unintended impacts on wetlands. More land managers are using conservation grazing as an alternative or supplement for burning to enhance upland prairie to provide and improve quality of wildlife habitat. Although much is still unknown about how conservation grazing is impacting wetlands, our study suggests that secretive marsh birds and ground-nesting waterfowl are not negatively affected by increased grazing frequency. If land managers in the tallgrass prairie are unable to burn as regularly as needed, our results indicate that the use of frequent, short duration grazing at high intensities to

manage upland cover likely has no or a positive effect on occupancy of secretive marsh birds and abundance of ground-nesting waterfowl pairs and broods. Additionally, the longer prairies go without a disturbance, the more dominated wetlands become by emergent vegetation, negatively affecting occupancy of some species of marsh birds. If land managers are considering secretive marsh birds in their management plans, we do not recommend leaving prairies idle for  $\geq 6$  years.



Figure 1. Grazing histories of sites surveyed in 2017 and 2018 in western Minnesota. Sites grouped by (a) years since last grazed and (b) the number of times grazed in the 3 years prior to being surveyed (grazing frequency).

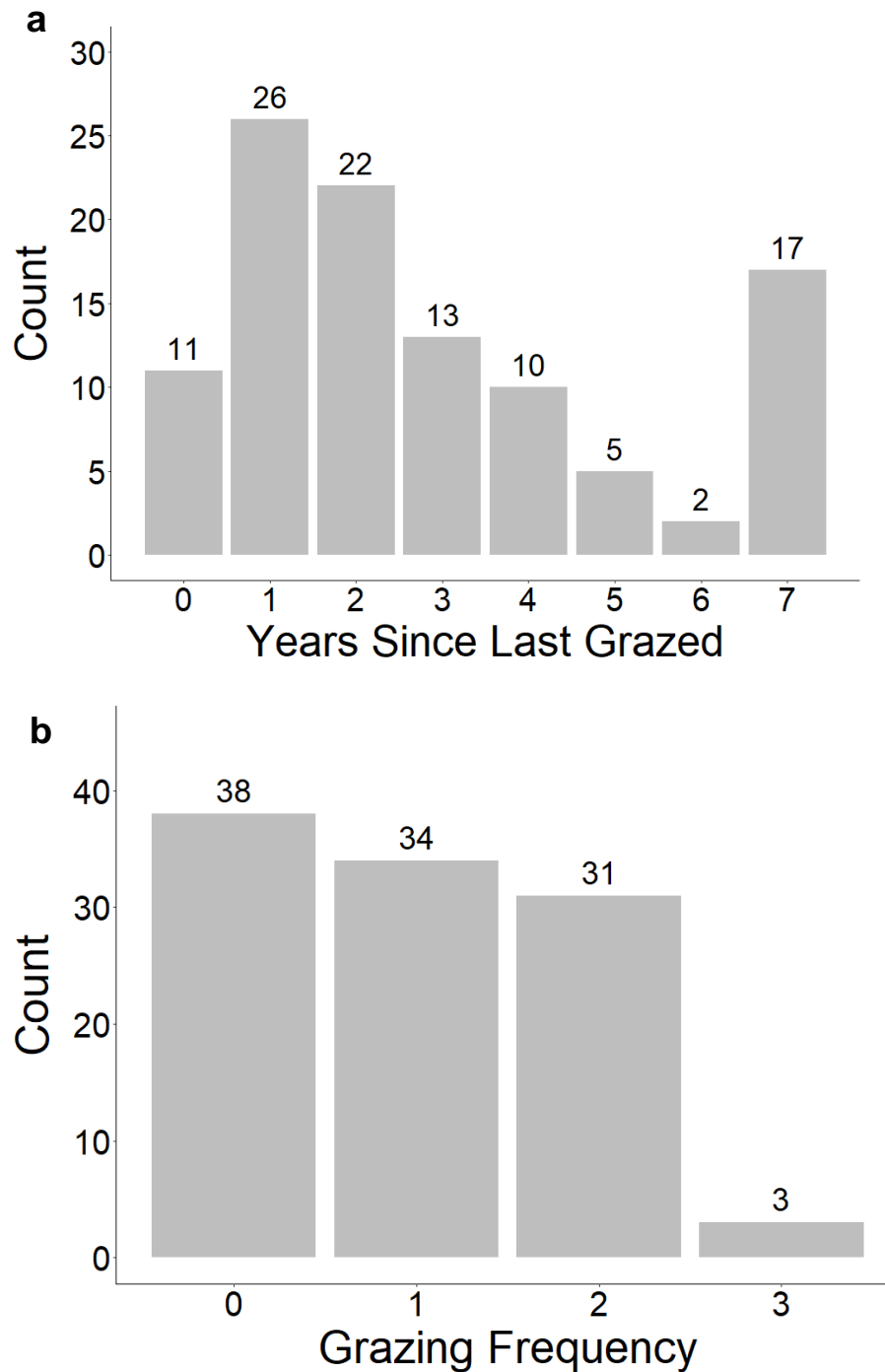


Table 1. Rankings of alternative linear models used to predict percent emergent vegetation in response to grazing history (grazing frequency in last 3 years or years since last grazed) and wetland permanence class in west central Minnesota, 2017-2018. analysis including number of parameters (K) and  $\Delta AIC$ .

<b>Model</b>	<b>K</b>	<b><math>\Delta AIC</math></b>
Grazing frequency + Wetland class	6	0
Years since grazed + Wetland class	6	2.12
Grazing frequency $\times$ Wetland class	10	3.18
Years since grazed $\times$ Wetland class	10	3.56
Wetland class	5	16.74
Grazing frequency	2	128.03
Years since grazed	2	132.61
Intercept-only	1	138.98

Figure 2. Predicted relationship between percent emergent vegetation and (a) grazing frequency (the number of times a site was grazed in the last 3 years) and (b) years since a site was last grazed, categorized by wetland permanence class, in western Minnesota's tallgrass prairie, 2017 – 2018. The shaded areas represent 85% confidence intervals.

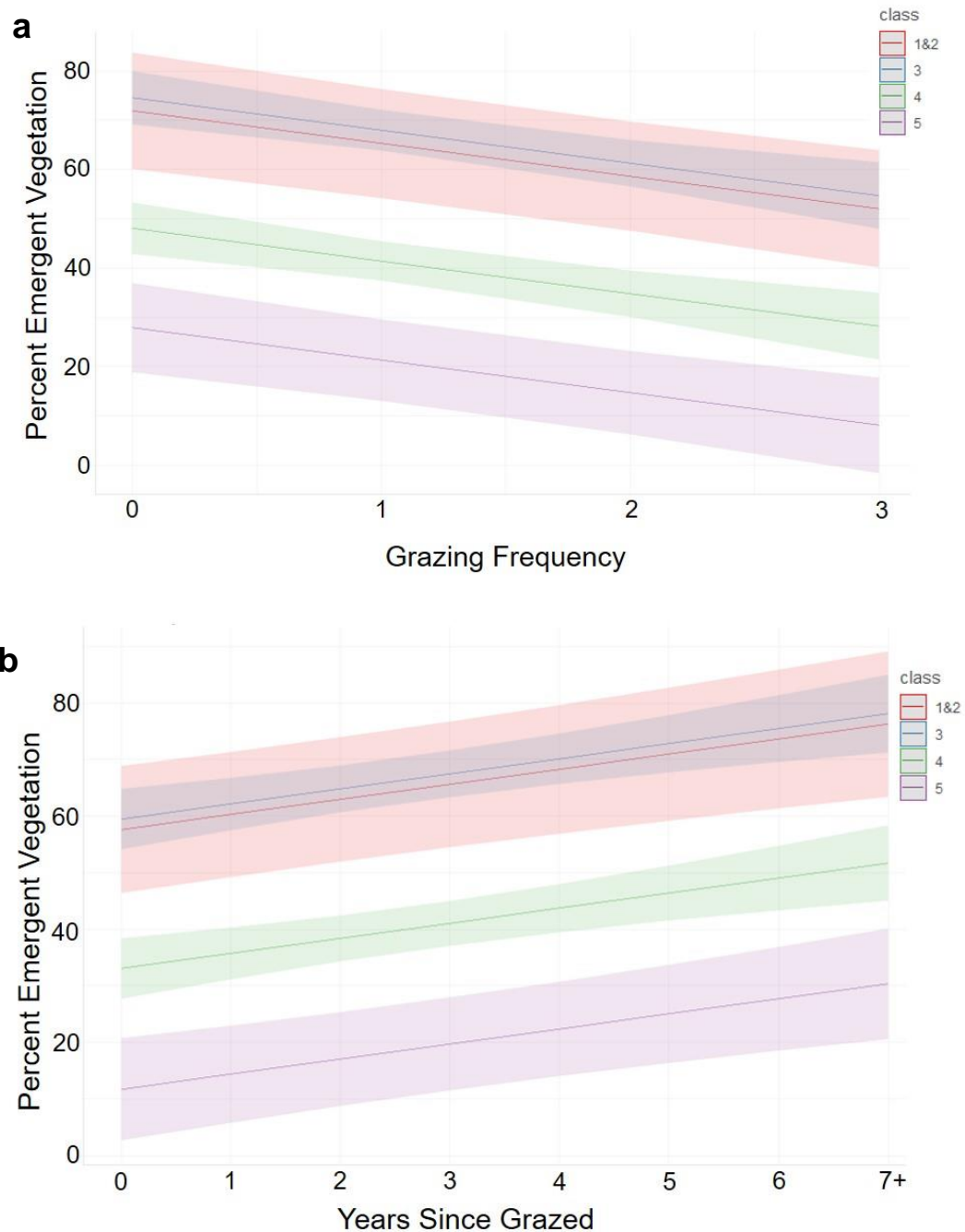


Table 2. Models of abundance of blue-winged teal and mallard indicated breeding pairs and abundance of blue-winged teal and dabbling<sup>a</sup> broods, based on surveys completed during the summers of 2017 and 2018 in western Minnesota's Prairie Pothole Region. Model results include number of parameters (K) and  $\Delta AIC_c$ .

Species	Model	K	$\Delta AIC_c$
Blue-winged Teal pairs	Base model <sup>b</sup>	15	0.00
	Base model + Years since grazed	16	1.37
	Base model + Grazing frequency	17	1.99
	Base model + Years since grazed <sup>2</sup>	16	3.34
Mallard pairs	Base model + Years since grazed	15	0.00
	Base model + Years since grazed <sup>2</sup>	16	0.30
	Base model	14	1.85
	Base model + Grazing frequency	15	2.71
All dabbling broods	Base model	12	0.00
	Base model + Years since grazed	13	1.31
	Base model + Grazing frequency	13	2.00
	Base model + Years since grazed <sup>2</sup>	14	3.29
Blue-winged Teal broods	Base model	12	0.00
	Base model + Years since grazed	13	1.58
	Base model + Grazing frequency	13	1.80
	Base model + Years since grazed <sup>2</sup>	14	3.36

<sup>a</sup>Insufficient observations of mallard broods prevented analysis of brood abundance, but we combined all dabbling broods together to estimate overall brood abundance in relation to grazing, <sup>b</sup>Base model included a suite of temporal, site, and landscape-level covariates: survey day, year, time of day, observer, longitude, wetland inundation, percent of emergent vegetation covering a wetland, area of surveyed wetland, wetland permanence class, number of years since a site was burned, the proportion of land cover classified as wetland habitat, and the proportion of land cover classified as agriculture.

Figure 3. Blue-winged teal (BWTE, a) and mallard (MALL, b) pair response to years since grazed, using the best-supported linear model of waterfowl abundance. Years since grazed was supported in the top model for mallard pairs but not blue-winged teal pairs. Lines represent mean predicted abundance, whereas dots represent predicted values for each survey site, given other unique combinations of covariates.

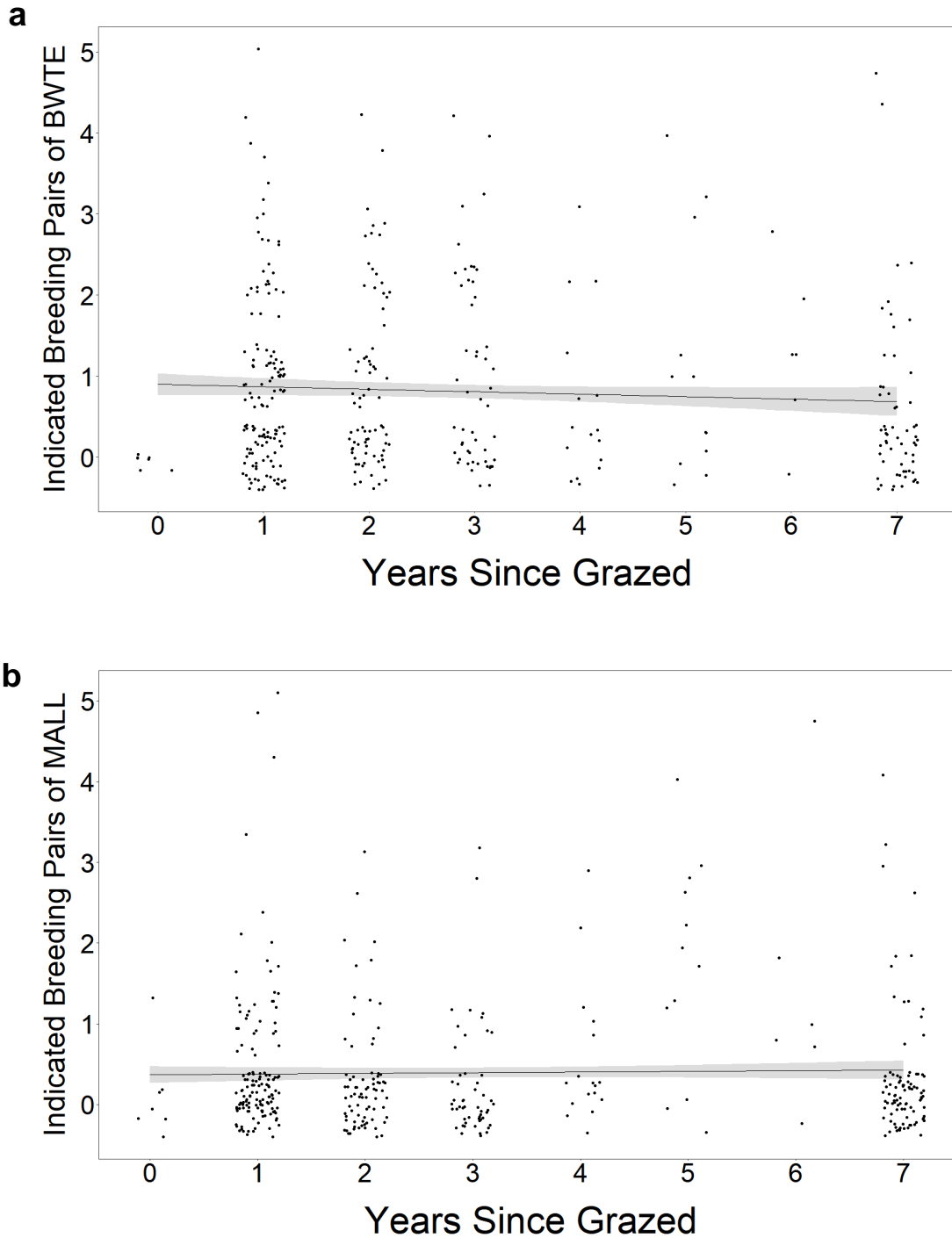


Figure 4. Blue-winged teal (BWTE, a) and mallard (MALL, b) pair response to grazing frequency (number of times grazed in the last three years). Grazing frequency was not supported in the top models for either blue-winged teal or mallard pairs. Lines represent mean predicted abundance, whereas dots represent predicted values for each survey site, given other unique combinations of covariates.

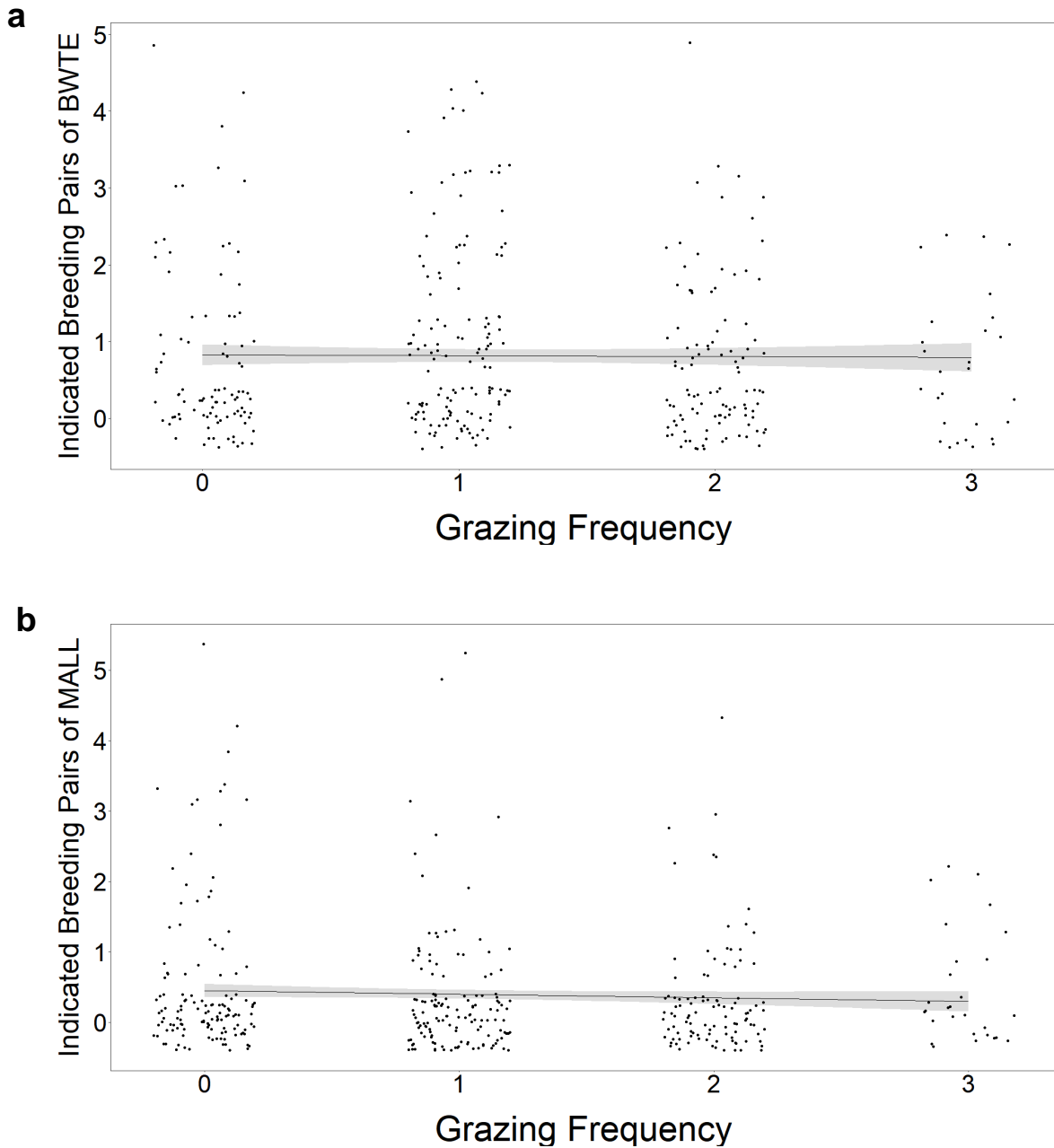


Figure 5. Vocalization probabilities across 11-minute survey of 4 species of secretive marsh birds: American bitterns (AMBI), pied-billed grebes (PBGR), soras (SORA), and Virginia rails (VIRA). Survey protocol consisted of 5 minutes of passive listening followed by 6 minutes during which we broadcasted calls of 6 species, each for 30 seconds followed by 30 seconds of passive listening.

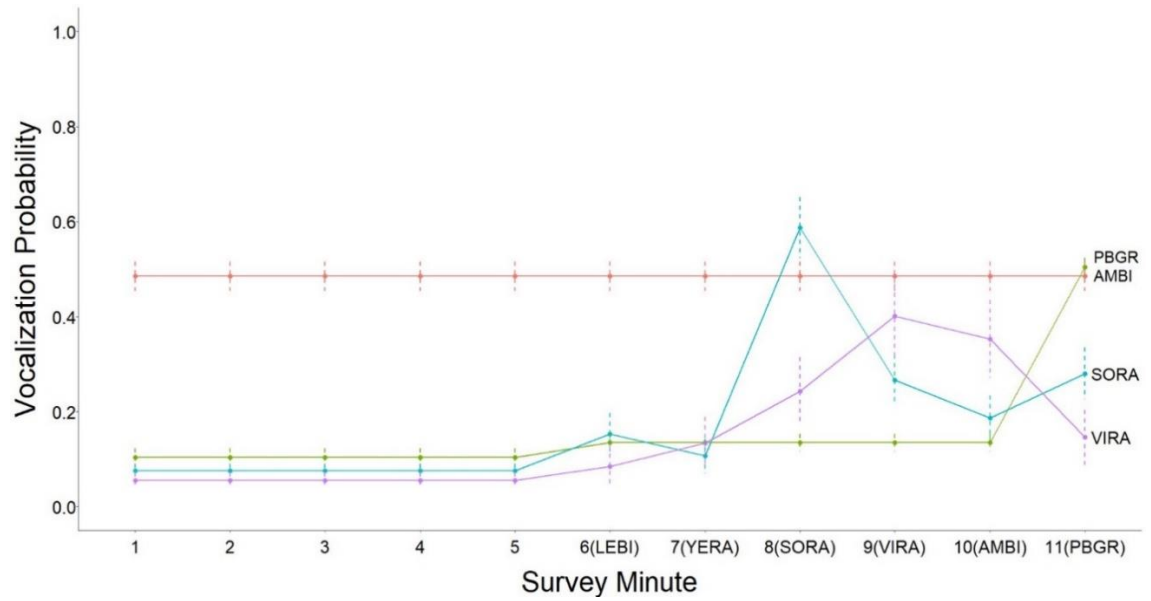


Table 3. Sequential model selection results for American bittern vocalization probability models (Stage 1)<sup>a</sup>, base occupancy model (Stage 2)<sup>b</sup>, and grazing models (Stage 3). Surveys completed during the summers of 2017 and 2018 in western Minnesota's Prairie Pothole Region. Model results include  $\Delta AIC_c$ , model weights ( $w_i$ ), number of parameters ( $k$ ), and deviance  $[-2\log(L)]$ .

<b>AMERICAN BITTERN</b>	<b><math>\Delta AIC_c</math></b>	<b><math>w_i</math></b>	<b><math>k</math></b>	<b><math>-2\log(L)</math></b>
<b>Stage 1: Model survey minute effects</b>				
Constant <sup>c</sup>	0.00	1.63	8	1024.12
Conspecific vs. heterospecific	0.97	0.62	10	1020.87
Conspecific	1.45	0.38	9	1023.47
Species-specific	8.37	0.00	14	1019.68
<b>Stage 2: Add detection and occupancy covariates</b>				
Base model	0.00	1.00	20	958.76
<b>Stage 3: Add grazing variables</b>				
Years since grazed <sup>d</sup>	0.00	0.64	21	951.77
Years since grazed <sup>2</sup>	0.66	0.33	22	950.15
Grazing frequency	3.25	0.03	21	955.02
Base model	4.73	0.01	20	958.76

<sup>a</sup>Vocalization probability models considered were 1) constant vocalization probability across 11-minute survey (Constant), 2) constant vocalization probability across entire survey, except for the minute of conspecific calling (Conspecific), 3) 1 vocalization probability during minutes of passive listening, 1 during the minute of broadcasting conspecific calls, and 1 for all 5 minutes of broadcasting heterospecific calls (Conspecific vs. heterospecific), and 4) 1 vocalization probability during passive minutes and 6 species-specific vocalization probabilities (Species-specific), <sup>b</sup>Base model included 6 covariates on detection probability (observer, noise level, AM vs. PM, time since sunrise/sunset, survey day, and year) and 5 covariates on occupancy (area of the surveyed wetland, percent of emergent vegetation covering the wetland, wetland permanence class, the number of years since a site was burned, and total wetland area within a 200-m buffer, <sup>c</sup> $AIC_c$  of top vocalization probability model was 1040.54, <sup>d</sup> $AIC_c$  of top grazing model was 996.55



Table 4. Sequential model selection results for pied-billed grebe vocalization probability models (Stage 1)<sup>a</sup>, base occupancy model (Stage 2)<sup>b</sup>, and grazing models (Stage 3). Surveys completed during the summers of 2017 and 2018 in western Minnesota's Prairie Pothole Region. Model results include  $\Delta AIC_c$ , model weights ( $w_i$ ), number of parameters ( $k$ ), and deviance  $[-2\log(L)]$ .

<b>PIED-BILLED GREBE</b>	<b><math>\Delta AIC_c</math></b>	<b><math>w_i</math></b>	<b><math>k</math></b>	<b><math>-2\log(L)</math></b>
<b>Stage 1: Model survey minute effects</b>				
Conspecific vs. heterospecific <sup>c</sup>	0.00	0.66	10	1410.73
Conspecific	0.84	0.28	9	1413.69
Species-specific	2.35	0.06	14	1404.48
Constant	96.72	0.00	8	1511.67
<b>Stage 2: Add detection and occupancy covariates</b>				
Base model	0.00	1.00	22	1383.16
<b>Stage 3: Add grazing variables</b>				
Base model <sup>d</sup>	0.00	0.73	22	1383.16
Years since grazed <sup>2</sup>	1.83	0.12	24	1380.39
Grazing frequency	2.26	0.08	23	1383.13
Years since grazed	2.30	0.07	23	1383.17

<sup>a</sup>Vocalization probability models considered were 1) constant vocalization probability across 11-minute survey (Constant), 2) constant vocalization probability across entire survey, except for the minute of conspecific calling (Conspecific), 3) 1 vocalization probability during minutes of passive listening, 1 during the minute of broadcasting conspecific calls, and 1 for all 5 minutes of broadcasting heterospecific calls (Conspecific vs. heterospecific), and 4) 1 vocalization probability during passive minutes and 6 species-specific vocalization probabilities (Species-specific), <sup>b</sup>Base model included 6 covariates on detection probability (observer, noise level, AM vs. PM, time since sunrise/sunset, survey day, and year) and 5 covariates on occupancy (area of the surveyed wetland, percent of emergent vegetation covering the wetland, wetland permanence class, the number of years since a site was burned, and total wetland area within a 200-m buffer, <sup>c</sup> $AIC_c$  of top vocalization probability model was 1040.54, <sup>d</sup> $AIC_c$  of top grazing model was 996.55

Table 5. Sequential model selection results for sora vocalization probability models (Stage 1)<sup>a</sup>, base occupancy model (Stage 2)<sup>b</sup>, and grazing models (Stage 3). Surveys completed during the summers of 2017 and 2018 in western Minnesota's Prairie Pothole Region. Model results include  $\Delta AIC_c$ , model weights ( $w_i$ ), number of parameters ( $k$ ), and deviance  $[-2\log(L)]$ .

<b>SORA</b>	<b><math>\Delta AIC_c</math></b>	<b><math>w_i</math></b>	<b><math>k</math></b>	<b><math>-2\log(L)</math></b>
<b>Stage 1: Model survey minute effects</b>				
Species-specific <sup>c</sup>	0.00	1.00	14	1652.24
Conspecific vs. heterospecific	12.58	0.00	10	1673.41
passive vs. playback	59.55	0.00	9	1722.50
Constant	198.82	0.00	8	1863.88
<b>Stage 2: Add detection and occupancy covariates</b>				
Base model	0.00	1.00	26	1608.39
<b>Stage 3: Add grazing variables</b>				
Years since grazed <sup>d</sup>	0.00	0.72	27	1592.36
Base model	1.40	0.18	26	1608.39
Years since grazed <sup>2</sup>	2.36	0.07	28	1592.36
Grazing frequency	2.92	0.04	27	1595.28

<sup>a</sup>Vocalization probability models considered were 1) constant vocalization probability across 11-minute survey (Constant), 2) constant vocalization probability across entire survey, except for the minute of conspecific calling (Conspecific), 3) 1 vocalization probability during minutes of passive listening, 1 during the minute of broadcasting conspecific calls, and 1 for all 5 minutes of broadcasting heterospecific calls (Conspecific vs. heterospecific), and 4) 1 vocalization probability during passive minutes and 6 species-specific vocalization probabilities (Species-specific), <sup>b</sup>Base model included 6 covariates on detection probability (observer, noise level, AM vs. PM, time since sunrise/sunset, survey day, and year) and 5 covariates on occupancy (area of the surveyed wetland, percent of emergent vegetation covering the wetland, wetland permanence class, the number of years since a site was burned, and total wetland area within a 200-m buffer, <sup>c</sup> $AIC_c$  of top vocalization probability model was 1040.54, <sup>d</sup> $AIC_c$  of top grazing model was 996.55

Table 6. Sequential model selection results for Virginia rail vocalization probability models (Stage 1)<sup>a</sup>, base occupancy model (Stage 2)<sup>b</sup>, and grazing models (Stage 3). Surveys completed during the summers of 2017 and 2018 in western Minnesota's Prairie Pothole Region. Model results include  $\Delta AIC_c$ , model weights ( $w_i$ ), number of parameters ( $k$ ), and deviance  $[-2\log(L)]$ .

<b>VIRGINIA RAIL</b>	<b><math>\Delta AIC_c</math></b>	<b><math>w_i</math></b>	<b><math>k</math></b>	<b><math>2\log(L)</math></b>
<b>Stage 1: Model survey minute effects</b>				
Species-specific	0.00	1.00	14	929.06
Conspecific vs. heterospecific	14.81	0.00	10	952.47
Conspecific	49.49	0.00	9	989.26
Constant	82.12	0.00	8	1023.99
<b>Stage 2: Add detection and occupancy covariates</b>				
Base model	0.00	1.00	26	910.68
<b>Stage 3: add grazing variables</b>				
Grazing frequency	0.00	0.58	27	907.77
Base model	0.57	0.33	26	910.68
Years since grazed <sup>2</sup>	2.55	0.05	28	907.96
Years since grazed	2.60	0.04	27	910.37

<sup>a</sup>Vocalization probability models considered were 1) constant vocalization probability across 11-minute survey (Constant), 2) constant vocalization probability across entire survey, except for the minute of conspecific calling (Conspecific), 3) 1 vocalization probability during minutes of passive listening, 1 during the minute of broadcasting conspecific calls, and 1 for all 5 minutes of broadcasting heterospecific calls (Conspecific vs. heterospecific), and 4) 1 vocalization probability during passive minutes and 6 species-specific vocalization probabilities (Species-specific), <sup>b</sup>Base model included 6 covariates on detection probability (observer, noise level, AM vs. PM, time since sunrise/sunset, survey day, and year) and 5 covariates on occupancy (area of the surveyed wetland, percent of emergent vegetation covering the wetland, wetland permanence class, the number of years since a site was burned, and total wetland area within a 200-m buffer, <sup>c</sup> $AIC_c$  of top vocalization probability model was 1040.54, <sup>d</sup> $AIC_c$  of top grazing model was 996.55

Figure 6. Predicted occupancy of marsh birds in response to years since grazed, using the best-supported linear (American bittern, sora) or quadratic model (pied-billed grebe, Virginia rail). Lines represent mean predicted occupancy, whereas dots represent predicted values for each survey site, given other unique combinations of covariates. Years since grazing effect was not included as a covariate in best-supported models of pied-billed grebe and Virginia rail occupancy, whereas best-supported models of occupancy of American bittern and sora included years since grazing.

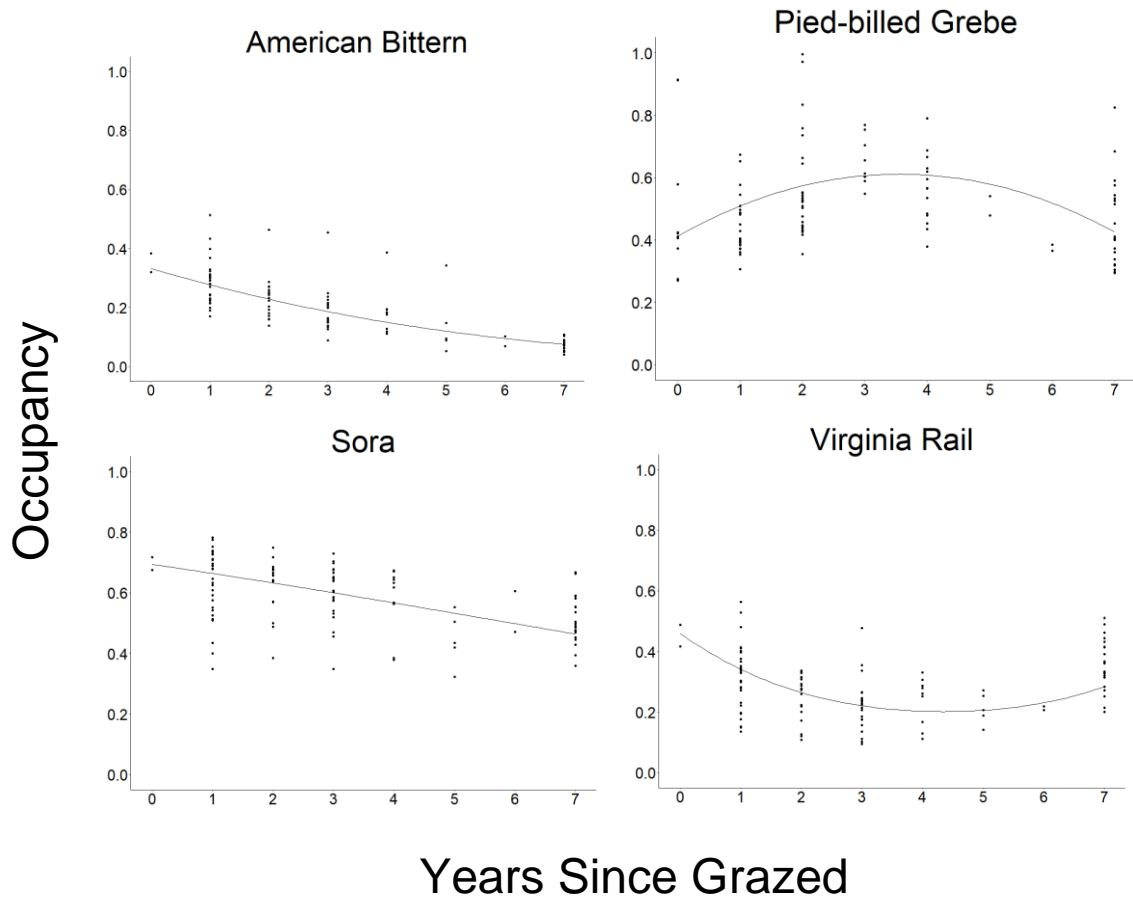
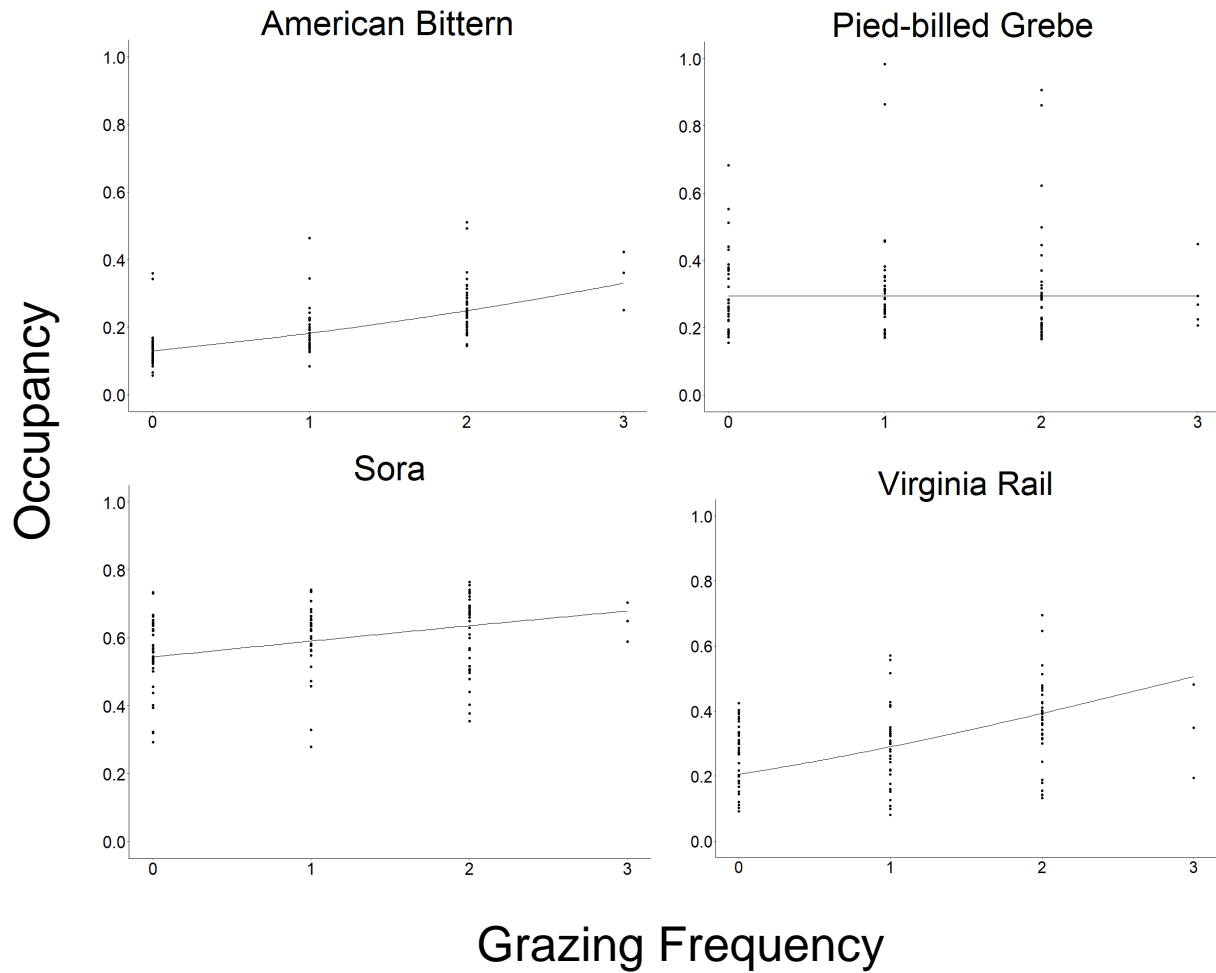


Figure 7. Predicted occupancy of marsh birds in response to grazing frequency (number of times grazed in the last 3 years). Lines represent mean predicted occupancy, whereas dots represent predicted values for each survey site, given other unique combinations of covariates. Grazing frequency was not included in the best-supported model for pied-billed grebe or sora, whereas grazing covariates were included in best-supported models of American bittern and Virginia rail occupancy.



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